

**THE FATE OF LANDFILL LEACHATE
IN WASTE WATER TREATMENT PLANTS
AND IN GROUNDWATER AT
ATTENUATION LANDFILLS**

FEBRUARY 1994



**Ministry of
Environment
and Energy**

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Final Report

Report prepared by:

Wilf Ruland and Shirley Schellenberg
Waterloo Centre for Groundwater Research, University of Waterloo

Grahame Farquhar
Department of Civil Engineering, University of Waterloo

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TABLE OF CONTENTS

1.1	Introduction	1
2.0	The Composition of Landfill Leachate	2
2.1	Chemical Characteristics of Leachate	2
2.2	Variations in Leachate Chemistry	4
3.0	Groundwater Attenuation of Landfill Leachate	6
3.1	Attenuation Processes	6
3.2	Research on Attenuation of Leachate Plumes	9
3.3	Critical Contaminants	10
3.4	Discussion	16
4.0	The Fate of Leachate in Sewage Treatment Plants	18
4.1	General Treatment Options	18
4.2	Characteristics of Leachate Affecting Treatment	19
4.2.1	Flow Variations	19
4.2.2	Leachate Composition	20
4.3	Effectiveness of Specific Treatment Processes	25
4.3.1	Biological Treatment Processes	25
4.3.2	Physical-Chemical Treatment Processes	27
4.4	Case Histories of Full-Scale Leachate Treatment	30
4.4.1	Highway 101 Landfill	30
4.5	Discussion	32
5.0	The Relative Merits of Using Groundwater Attenuation versus Using Waste Water Treatment Plants to Deal with Landfill Leachates	34
6.0	Conclusions and Recommendations	36
7.0	References	39
8.0	Appendices	
	Appendix A: Terms of Reference of Study	53
	Appendix B: MOE Reasonable Use Policy	61
	Appendix C: Suggested Guidelines for Attenuation Landfill Sites	87
	Appendix D: Case Studies of Four Landfills in Ontario	92
	Appendix E: Summaries of Full-Scale Leachate Treatment Systems	96

LIST OF TABLES

Table 1:	Representative Ranges for Inorganic Parameters In landfill Leachate	2
Table 2:	Commonly Identified Organic Contaminants in Plumes of Groundwater Contaminated by Landfill Leachate	3
Table 3:	Typical Landfill Leachate Contaminant Concentrations at Different Ages	23
Table 4:	Guidelines for Selecting Aerobic Biotreatment Processes	28

LIST OF FIGURES

Figure 1:	Possible Treatment Process Train to Achieve Steam Quality FEDC for a Young Landfill Leachate	21
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1.0 INTRODUCTION

A landfill is a place where wastes are disposed of by burial under layers of soil. Wastes are trucked into a landfill site, are placed in the working area of the site and compacted, and are covered with soil at regular intervals. In Ontario, landfilled wastes must be covered at the end of each working day with a 15 cm thick layer of clean soil.

Landfills are the primary method of waste disposal in Ontario. In Ontario the Ministry of the Environment and Energy (MOEE) is responsible for regulating the establishment, operation, and closure of landfills (MOEE, 1985). Considerable time and effort are invested by MOEE personnel in trying to ensure that environmental impacts associated with landfills are minimized and mitigated.

The most troublesome environmental impacts of landfills are usually caused by leachate. Leachate is a contaminated liquid which is generated at almost all landfills (Freeze and Cherry, 1979). It forms when precipitation seeps into a landfill, and "leaches" chemicals from the buried wastes.

Landfill leachate can contain thousands of chemicals, many of which may be hazardous (Cherry et al, 1987). Landfills where leachate is untreated and uncontrolled can cause severe contamination of both groundwater and surface water. A crucial design decision for every landfill, which must be reviewed by the MOEE, involves determining the most appropriate method of dealing with the leachate.

Landfills can be classified into two basic types, depending on the leachate treatment option being employed at the landfill:

Containment Landfills

At "containment" landfills as much as possible of the leachate is contained within the site and collected, and then treated at a sewage treatment plant (WWTP). After treatment, the WWTP effluent is discharged to a nearby surface water body.

Attenuation Landfills

At "attenuation" landfills, migration of leachate from the landfill into the surrounding groundwater system is allowed as part of the design. The natural attenuating mechanisms of the local groundwater flow system are relied upon to treat the leachate from such landfills.

It is implicit in these definitions that containment landfills have a greater impact on surface waters (because this is where WWTP effluent is discharged), while attenuation landfills have a greater impact on groundwater (since the groundwater is being relied upon for leachate treatment).

The decision on how best to design a landfill (including how to best treat the leachate from a landfill at a given location) is often very difficult and complex. The purpose of this study is to give an overview of the factors to be considered in the decision-making process as it pertains to the two methods of leachate treatment, and to provide a discussion of the relative merits and drawbacks of the two methods of leachate treatment.

The opinions and recommendations provided in this paper are those of the authors. They do not necessarily reflect MOEE policies. The full terms of reference of the study are included in Appendix A.

2.0 THE COMPOSITION OF LANDFILL LEACHATE

Knowledge of leachate composition plays a very important role in the design decision of how to best treat a landfill's leachate. For example, if a landfill's leachate is very weak then it may not matter how the leachate is treated. On the other hand, at landfill sites where it is necessary to protect the groundwater from contamination if a leachate is very strong and the attenuative capacity of the local subsurface environment is low then there may be no choice but to collect and treat the leachate at a WWT.

2.1 The Chemical Characteristics of Landfill Leachates

Landfill leachates are usually strong (total dissolved solids are typically thousands of milligrams per Litre) and very complex mixtures of thousands of contaminants. Some contaminants may have been leached directly from the wastes in a landfill, while others may have formed as a result of chemical reactions or biological processes taking place within the landfill.

Contaminants in leachate can be classified into six categories, on the basis of their reaction types and modes of occurrence: trace metals, nutrients, other inorganic species, organic contaminants, biological contaminants, and radioactive contaminants (Domenico and Schwartz, 1990).

Trace metals, nutrients, and other inorganic species in leachate are often grouped together as inorganic contaminants. Driscoll (1986) has prepared a table with typical ranges of inorganic contaminants in landfill leachates. The table is based on work done by Griffin et al (1976) and Leckie et al (1975), and is summarized in Table 1 below.

Table 1: Representative Ranges for Inorganic Parameters in Landfill Leachate

<u>Parameter</u>	<u>Representative Range (mg/L)</u>
total dissolved solids	5000 to 40,000
potassium	200 to 1000
sodium	200 to 1200
calcium	100 to 3000
magnesium	100 to 1500
chloride	300 to 3000
sulphate	10 to 1000
alkalinity	500 to 10,000
iron	1 to 1000
manganese	0.01 to 100
nickel	0.01 to 1
zinc	0.1 to 100
nitrate	0.1 to 10
ammonia	10 to 1000
organic nitrogen	10 to 1000
total dissolved organic carbon	200 to 30,000
chemical oxygen demand (COD)	1000 to 90,000

(Sources: Driscoll, 1986; after Griffin et al, 1976 and Leckie et al, 1975)

Leachates will generally have elevated levels of trace metals such as iron, nickel, zinc and lead. Common nutrients which are at elevated levels in leachates include ammonia, nitrate, and phosphorous compounds. Other inorganic chemicals typically found at elevated concentrations in leachates include major ions such as sodium, potassium, calcium, magnesium, chloride, and sulphate.

In addition to inorganic chemicals, landfill leachates contain many organic chemicals. Elevated levels of general organic parameters such as BOD are typical for landfill leachates, and indicate that numerous organic compounds must be present. Tables of the most commonly detected organic chemicals in landfill leachates have been prepared by authors such as Feenstra (1987) and Reinhard et al. (1984). A number of such tables have been summarized in Table 2 of this paper (below).

Table 2: Commonly Identified Organic Contaminants in Plumes
of Groundwater Contaminated by Landfill Leachate

Aromatic Hydrocarbons

benzene
toluene
xylenes
ethylbenzene

Carboxylic Acids

many are present, but are
not commonly identified

Polynuclear Aromatic Hydrocarbons

napthalene
others may be present, but
are not commonly identified

Phenolic Compounds

phenol
others may be present, but
are not commonly identified

Halogenated Hydrocarbons

Aliphatics:

carbon tetrachloride
perchloroethylene
trichloroethylene
dichloroethylenes
vinyl chloride
chloroform
dichloromethane
1,1,1 trichloroethane
dichloroethanes
chloroethane

Aromatics:

chlorobenzene
dichlorobenzenes

Sources: Barker et al (1987), Feenstra (1987), Cherry et al (1985),
Reinhard et al (1984)

The most commonly identified organic chemicals in landfill leachates are typically aromatic hydrocarbons found in petroleum such as benzene, toluene, and xylenes and chlorinated hydrocarbons such as trichloroethylene, 1,1,1-trichloroethane and chlorobenzene.

Modern Ontario landfill leachates may also contain biological contaminants such as bacteria and viruses, and radioactive contaminants. Such contaminants are typically not tested for in leachates, and there is little if any information about such contaminants in leachate in the published literature.

2.2 Variations in Leachate Chemistry and in Leachate Generation Rates

Leachate chemistry varies considerably from one landfill to another. Reitzel et al (1992) list factors which influence the composition of a landfill's leachate, including the following:

- i) the types of waste buried in the landfill;
- ii) the nature of the landfill operation;
- iii) the local climate;
- iv) local hydrogeologic conditions;
- v) conditions within the landfill such as moisture content, temperature, pH and chemical and biological activity.

At any given landfill the leachate composition may also vary considerably. Leachate quality at a landfill can vary spatially (from one point to another within the landfill) and temporally (over time).

Spatial variations in leachate quality can be caused by changes in the composition of waste across a landfill. Spatial variations in leachate quality can also be caused by the different physical, chemical, and biological conditions at different locations within the landfill. Finally, non-aqueous phase liquids (NAPLs) such as petroleum products or chlorinated solvents can lead to very significant spatial variations in leachate quality if they are present as wastes in a landfill.

Temporal variations in leachate quality have been described by many authors, including Reitzel et al (1992) and McGinley and Kmet (1984). Concentrations of most contaminants in leachate generally rise rapidly and reach a maximum shortly after the refuse has become saturated (McGinley and Kmet, 1984). This is followed by a steady decline in leachate strength to relatively stable, long-term, low levels of contaminants in the leachate. The rate of decline varies for different parameters; for example, the rate of decline for BOD is more rapid than for iron or ammonia.

Temporal variations in leachate strength are important from a regulatory perspective, because these variations will cause the potential environmental impacts of a landfill to also vary over time. This is recognized by the MOEE, which defines the contaminating lifespan of a landfill as "the period of time during which the landfill will produce contaminants at levels that

could have (an) unacceptable impact if they were discharged into the surrounding environment" (MOEE, 1988). The MOEE requires that the service life of engineered facilities at a landfill exceed its contaminating lifespan.

In addition to the variations in leachate chemistry outlined above, there are also significant variations in rates of leachate generation at landfills. Variations in leachate generation rates are directly related to the size (ie. the areal extent) of a landfill, but will also be affected by the following factors:

- fluctuations in precipitation;
- the types of daily, intermediate, and final cover used at the landfill;
- the extent to which completed portions of the landfill receive intermediate or final cover;
- the slopes of the top and sides of the landfill;
- groundwater flow through the landfill (if this is occurring);
- surface water management procedures at the landfill.

The strength and chemical complexity of most landfill leachates, coupled with the temporal and spatial variations in their chemistry and in rates of their generation, may pose great difficulties when trying to determine how to deal with them. Additional complexity is introduced into the decision-making process because the fate of leachate in the groundwater flow system at attenuation landfills is often not well understood, and because many WWTPs require costly upgrades to be able to adequately treat leachate. The next sections of this paper present a much more detailed discussion of these issues.

3.0 GROUNDWATER ATTENUATION OF LANDFILL LEACHATE

At attenuation landfill sites leachate is allowed to migrate out of the landfill, through the unsaturated zone (if the landfill is above the water table), and into the saturated groundwater zone. Attenuation sites work on the principle that as the leachate plume moves through the groundwater flow system the concentration of chemicals in the leachate will be reduced by natural processes, so that at some critical distance from the landfill concentrations will be at or below acceptable levels. This reduction in concentration is referred to as attenuation.

The main processes of attenuation include dilution, adsorption, biodegradation, exchange reactions, precipitation, and filtration. The effectiveness of each of these processes in attenuating chemicals in the leachate depends on several factors, including the physical and chemical properties of a given chemical and the characteristics of the geologic materials and the groundwater flow system.

As leachate travels through the ground it encounters varying physical and chemical conditions, which cause the composition of the leachate plume in the groundwater to continuously change. Some chemicals drop out of the leachate and are left behind adhering to the soil material, some are transformed into different chemicals, and others simply become more dilute by mixing with the surrounding groundwater.

The leachate-derived groundwater contamination plume may also pick up additional chemicals from the soil particles it flows through, and over time chemicals that had at one point adhered to the soil particles may come free and rejoin the leachate plume.

3.1 Attenuation Processes

The following discussion of the various attenuation processes that may be at work within a leachate plume is based on Domenico and Schwartz (1990) and Bagchi (1987).

Dilution or Hydrodynamic Dispersion

Dilution is a process by which the concentration of leachate chemicals is reduced through mixing with surrounding groundwater. The mixing process which causes dilution is referred to as hydrodynamic dispersion. Hydrodynamic dispersion (or dilution by mixing) is caused by a combination of two processes: diffusion and mechanical dispersion (Domenico and Schwartz, 1990).

Diffusion is a relatively slow process caused by the random molecular motion of chemicals dissolved in water. The process causes chemicals to move in the direction of their concentration gradients from areas of higher to areas of lower concentration. It causes chemicals, which are at a higher concentration in leachate than in background groundwater, to slowly mix with and move toward equilibration with background groundwater. Diffusion becomes significant as an attenuation process in flow systems where groundwater movement is very slow. Mechanical dispersion is a more mechanical mixing process which is caused by

groundwater moving at different speeds and in different directions as it travels along various flow paths through a geologic deposit. Both diffusion and mechanical dispersion result in the spreading of the leachate plume as the plume moves downgradient through the groundwater flow system. At most attenuation landfills groundwater velocity is high enough that mechanical dispersion would account for a much greater amount of plume spreading than diffusion.

Some parameters in municipal landfill leachates such as chloride are attenuated only by dilution. The major factors that affect dilution are density differences between leachate and groundwater, the rate of movement of the leachate plume through the groundwater flow system, the diffusion coefficient of the leachate chemicals, the dispersion coefficients of the aquifer, and soil stratigraphy.

Adsorption

Adsorption of chemicals may occur in the presence of solid organic matter in a geologic deposit. Adsorption can retard the spread of some contaminants or even immobilize them (Domenico and Schwartz, 1990) causing a lowering of total dissolved solids (TDS) and organic contaminant concentrations in the leachate plume. For example, adsorption in the presence of either soil organic carbon matter or clay minerals has been recognized as an important attenuative mechanism for dissolved non-polar hydrophobic organic molecules.

The adsorption of leachate-derived chemicals in groundwater onto the surfaces of solid organic matter or clay minerals in a geologic deposit is dependent on the particles' surface capacity, and is affected by pH. Once the surface capacity is filled, additional chemicals can not sorb on unless they displace an already sorbed chemical. If that occurs, the desorbed chemical will re-enter the groundwater (Giles, 1970).

An important and sometimes overlooked aspect of adsorption is its temporary nature as an attenuative mechanism. Adsorption reactions may result in significant retardation of the rate of movement of some organic chemicals in a leachate plume, but the adsorption reactions are reversible (Barker, 1987).

Biodegradation and Biological Uptake

Biodegradation (or biotransformation) is a microbially catalyzed reaction that usually involves the transformation of organic compounds into simpler inorganic forms, but may result in transformation into simpler organic forms (Domenico and Schwartz, 1990). The transformation products may not have originally been present in the leachate plume. In landfills, biodegradation is carried out mainly by anaerobic bacteria resulting in methane gas production (Bagchi, 1987).

Biological uptake involves adsorption or other types of retention of leachate-derived chemicals by micro-organisms. Biological uptake is not considered permanent, because as leachate strength and thus food supply for the micro-organisms decreases their population declines and chemicals fixed to those cells may then be released.

Exchange Reactions

Exchange reactions mainly occur with clay minerals, and involve the exchange of ions of one type by another type without disturbing the clay mineral structure (Bagchi, 1987). The most important natural exchange reactions see Ca^{2+} and Mg^{2+} cations being taken out of groundwater, and replaced by Na^+ ions which are available from the surfaces of some clay minerals (Domenico and Schwartz, 1990).

Because a simple exchange is involved, the TDS in leachate-contaminated groundwater is not lowered as a result of exchange reactions. Cation exchange capacity increases as the pH of the system increases, and cation exchange occurs mostly with hydrated ions. Anion exchange capacity increases as the pH of the system decreases.

In low pH systems that have clayey soils present, negatively charged organic ions will be attenuated through anion exchange (Bagchi, 1987). In southern Ontario, where the pH of most Ontario leachate-soil systems approaches a neutral value, significant attenuation of organic chemicals by this process is not expected.

Precipitation

Precipitation involves a phase change whereby dissolved species combine to form solid phase compounds which may subsequently be deposited on the soil matrix. This occurs when the concentration of the dissolved species exceeds its solubility limit (Bagchi, 1987). Solubility limits depend mainly on redox potential and pH.

Precipitation of leachate-derived chemicals, usually metals, will occur as a result of changes in the chemical characteristics of the groundwater flow system through which a leachate plume is travelling. A good example of the attenuation of metals by precipitation is provided by the Babylon Landfill (Kimmel and Braids, 1980). The pH of groundwater is in the 6 to 6.5 range, and both iron and manganese are mobile in the moderately reducing conditions found near the landfill. As the plume moves downgradient into more oxidizing conditions, a dramatic reduction in the mobility of these metals is observed.

The opposite process to precipitation is dissolution where solid phases dissolve and reenter solution. Under reducing redox conditions, a plume may take up metals (such as manganese and iron) into solution through dissolution.

Filtration

Filtration is the process by which suspended solids in leachate are physically trapped in the pore structure of the soil system. In cases where chemical precipitation, biological growth, and other processes produce solid particles from leachate-derived chemicals, filtration of those particles may be a significant attenuation process.

3.2 Research on Attenuation of Leachate Plumes

The attenuation processes listed above can be demonstrated in the laboratory, and it is suspected that they will also occur to some extent in natural settings. It may, however, be very difficult to demonstrate that a specific attenuation process has actually occurred in a natural setting because in natural settings it is not possible to control variables to the extent that is possible in laboratory settings.

If a chemical is present in a landfill's leachate and is no longer present in the leachate plume at a given distance from the landfill, it is assumed that the chemical has been attenuated and one may speculate about the most probable attenuation mechanism. However, further studies are then needed before it is possible to confirm which mechanism was responsible for the observed attenuation. Until attenuation mechanisms are better understood it will be difficult to make accurate generalizations about attenuation processes which would be applicable in different hydrogeological settings.

There are several types of studies in the literature that look at leachate plumes generated by municipal landfills. There are studies of a general nature that attempt to define the size and shape of a plume, identify the major constituents in the plume, and assess the rate at which the plume is moving. Other studies attempt to observe the attenuation processes that are at work within the plume. These studies usually focus on a very small number of chemicals, and address only one or two attenuation processes. For instance, to study the process of hydrodynamic dispersion, a conservative tracer such as chloride is usually monitored.

The effectiveness of attenuation processes in natural settings is not well understood, and requires considerable additional research. There are, however, some significant findings with respect to certain types of chemicals and hydrological settings, and many of these are discussed in this paper.

Although it is difficult with today's knowledge to determine which attenuation processes are at work in a given groundwater flow system at a particular landfill, one way to assess the sum of the attenuation processes is to compare the chemical composition of leachate to the composition of the contaminant plume at increasing distances from the landfill. This method allows one to identify chemicals that may be mobile and persistent in a given groundwater flow system, but it can only be applied to already-existing landfills.

3.3 Critical Contaminants

A given chemical can be defined as a *critical contaminant* at a landfill site if it has all of the following characteristics:

- the chemical is present in the leachate at levels significantly greater than the Reasonable Use Policy (RUP) limit for that chemical. The Ministry of the Environment's Reasonable Use Policy (RUP) is outlined in Appendix B;
- the chemical is mobile in the local groundwater flow system;
- the chemical is persistent in the local groundwater flow system.

The determination of whether or not a chemical is considered to be a critical contaminant is thus dependent not only on the properties of the chemical and on the physical and chemical characteristics of the groundwater flow system in the landfill site area. It is also dependent upon the presence or absence of regulatory standards for the chemical, which in turn are based on the available information with respect to the health and aesthetic impacts of the chemical in groundwater. The critical contaminant at a given landfill may change over time, as the landfill matures and its leachate composition changes.

As mentioned earlier, contaminants (including leachate-derived chemicals) can be classified into six categories (Domenico and Schwartz, 1990):

- 1) trace metals
- 2) nutrients
- 3) other inorganic species
- 4) organic contaminants
- 5) biological contaminants, and
- 6) radioactive contaminants

The following discussion considers these groups of chemicals and identifies chemicals or substances within each group which are likely to be critical contaminants in landfill leachates. The discussion is based on the case studies and investigations available in the published literature.

1) Trace Metals

Landfill leachate commonly contains significant concentrations of trace metals, but these are generally not mobile at the neutral pH levels found in southern Ontario groundwaters. As a result most toxic trace metals (such as cadmium, chromium, and lead) are not usually considered critical contaminants because they are not commonly found in leachate plumes at levels exceeding Reasonable Use Policy limits. Metals may however pose significant problems if leachate springs are discharging to surface waters at the perimeter of the landfill.

Two metals often exceed Reasonable Use Policy limits - iron (ODWO of 0.3 mg/l) and manganese (ODWO of 0.05 mg/l). Iron and manganese are usually present in leachate at elevated concentrations, and are often mobile under the anaerobic groundwater conditions near landfills. Even if they are not present in a landfill, they may be incorporated

into the leachate plume due to their dissolution from soil particles as the plume moves through the groundwater flow system (Cherry, 1983). There are, however, many areas where natural background levels of iron and magnesium exceed the ODWO and in these areas the Reasonable Use Policy does not apply for these metals.

2) Nutrients

Nutrients include those ions or organic compounds containing nitrogen or phosphorous (Domenico and Schwartz, 1990). Phosphorous is not a critical contaminant in leachate plumes because it has only limited mobility and because it is not a focus of regulatory concern.

Nitrogen compounds (which include nitrate, nitrite, ammonia, and organic nitrogen) are usually present within landfills and landfill leachate plumes. Ammonia (in the form of the ammonium ion) is often the dominant nitrogen compound under the anaerobic conditions found within landfills, and ammonia may exceed regulatory limits if leachate enters surface waters near the landfill.

With increasing distance from the landfill, transformation of ammonia to nitrate may occur as mixing brings oxygen into the contaminant plume (Kimmel and Braids, 1980). The ODWO for nitrate is 10 mg/l, and it is important to consider that nitrate may exceed Reasonable Use Policy (RUP) limits downgradient of the landfill even if it is not present at elevated levels in the landfill leachate.

3) Other Inorganic Species

This group of chemicals includes the common metals (calcium, magnesium, sodium and potassium), ions containing sulphur and carbon, and anions such as chloride and bromide (Domenico and Schwartz, 1990).

Of the major ions, chloride is the only chemical considered a critical contaminant. The ODWO for chloride is 250 mg/l, and it is commonly found in landfill leachates at concentrations exceeding 1000 mg/L. Chloride is found in all leachate plumes, and since it is not affected by chemical or biological attenuation mechanisms the only attenuation which occurs is by dilution due to mechanical dispersion and diffusion. As a result, it is often used as a conservative tracer of landfill leachate contamination.

Calcium, magnesium, sodium, and potassium are found in landfill leachate plumes and are relatively mobile. They are generally not considered to be critical contaminants because there are no ODWO set for these metals and thus the RUP usually does not apply.

Sulfate is seldom present at significant levels in leachate plumes due to the strong reducing conditions found within and near landfills (Cherry, 1982).

4) Organic Chemicals

The relative decline in dissolved organic carbon along the length of a plume is not much different than the relative decline of a conservative

parameter such as chloride. This suggests that the gross load of organic matter in the plume is affected mainly by dilution due to mixing and may not be strongly influenced by adsorption and microbial degradation.

Specific organic compounds may however be attenuated by mechanisms besides dilution along the plume (Cherry et al, 1981). Barker et al (1987) indicate that two other processes are found to be significant in lowering organic concentrations in leachate plumes - biotransformation and sorption. The organic chemicals which are critical contaminants will thus be those organic compounds which are commonly found at high concentrations in leachate relative to their RUP limits, and which are least affected by the attenuative mechanisms outlined above.

Organic chemicals found in leachate can be divided into sixteen main classes, based on the functional groups that give them their 'group' chemical and physical properties (Domenico and Schwartz, 1990).

Out of these sixteen classes, there are five classes which are of particular interest when one considers landfill leachates:

- a) halogenated hydrocarbons (including both the aliphatic and aromatic subclasses)
- b) carboxylic acids
- c) phenols
- d) polynuclear aromatic hydrocarbons, and
- e) aromatic hydrocarbons.

Of these categories, most of the research in the literature focuses on halogenated hydrocarbons and the aromatic hydrocarbons.

a) Halogenated Hydrocarbons

This class is large and includes two major subclasses - the aliphatic halogenated hydrocarbons and the aromatic halogenated hydrocarbons (Domenico and Schwartz, 1990).

Critical contaminants in the aliphatic subclass may include chlorinated solvents such as trichloroethylene, industrial chemicals such as vinyl chloride, and pesticides such as aldrin and dieldrin. Critical contaminants in the aromatic subclass may include industrial chemicals such as chlorobenzenes.

Aliphatic halogenated hydrocarbons are often quite mobile in groundwater, but some can be attenuated by biodegradation (Barker and Cherry, 1986). The processes are not yet well understood but biotransformation has been documented for aliphatic halogenated hydrocarbons, mainly under anaerobic conditions. For example, at the North Bay landfill, biotransformation under strictly anaerobic conditions has probably caused the aliphatics 1,1,1-trichloroethane and trichloroethylene (TCE) to be restricted to the immediate vicinity of the landfill (Barker et al, 1986).

The biotransformation of aliphatics can lead to the production of "breakdown products", which in some cases may be more problematic

than the original contaminants. A well known transformation or "breakdown" sequence is tetrachloroethylene -> trichloroethylene -> dichloroethylene -> vinyl chloride, which has been documented to occur under anaerobic conditions by Vogel and McCarty (1985).

Biotransformation leads to the loss in the plume of the original organic chemical, but also leads to increased levels in the plume of its breakdown product(s). This may be important for two reasons:

- it may complicate monitoring of the plume, because there may be breakdown products found in the plume which are not present in the landfill leachate;
- it may result in increasingly mobile aliphatics being present in the plume, since aliphatics become increasingly hydrophilic and thus increasingly mobile as one moves down the breakdown sequence (Jackson et al, 1985).

Barker et al (1986) suggests that dispersion and retardation by adsorption may also limit aliphatic migration in leachate plumes. At the New Borden landfill, TCE is present at 750 ug/l in leachate but is only present at less than 1 ug/l 50 m from the landfill. At this site TCE is more highly sorbed than carbon tetrachloride.

Aromatic halogenated hydrocarbons most likely to be found in landfill leachates are the chlorinated benzenes. Many of the chlorinated benzenes have relatively low Reasonable Use Policy limits, and some have been shown to be quite mobile in groundwater (Barker et al, 1984).

In the aromatic subclass of halogenated hydrocarbons the PCBs as a group have received the most widespread public, media, and regulatory attention. PCBs should not be a focus of concern from the perspective of landfill leachate plume migration in groundwater, because they are relatively immobile in groundwater.

b) Aromatic Hydrocarbons

Aromatic hydrocarbons are a major constituent of petroleum products, and include the "BTEX" chemicals (benzene, toluene, ethylbenzene, and xylenes). Benzene has the lowest RUP limit and appears to be quite mobile, thus it is the aromatic hydrocarbon most likely to be a critical contaminant.

Aromatic hydrocarbons appear to be affected by biodegradation (Barker and Cherry, 1986) and adsorption (Barker et al, 1986). Aromatic hydrocarbons are readily biodegraded in the presence of oxygen in sandy unconfined aquifers, so that generally once such plumes become aerobic the rapid biodegradation of aromatics can be expected (Barker et al, 1987).

On the other hand, aromatic hydrocarbons can be quite mobile and persistent if the contaminant plume remains anaerobic. For example, in North Bay the municipal landfill is located on a sandy

unconfined aquifer and the flow system is anaerobic. Non-transformed organic contaminants consisting mainly of aromatic hydrocarbons have spread throughout the 700 m flow system.

These contaminants were transported for such a distance mainly because of the anaerobic conditions, the high flow velocity (75 m/yr) and the low organic sorption properties of the aquifer (Barker et al, 1986). The "case history" of the North Bay landfill in Appendix C provides a more detailed description of the landfill, the local hydrogeology, and the plume characteristics.

Further investigations into the behaviour of aromatic hydrocarbons within the leachate plume have been more recently undertaken at the North Bay landfill (Acton and Barker, 1992). Based on these studies, combined with laboratory investigations, it appears that some anaerobic biodegradation of aromatic hydrocarbons occurs within the leachate plume. Toluene was found to be most quickly degraded, and benzene was most persistent in the aquifer.

c) Phenolic Compounds

Phenols are compounds which are common in nature, but today some are also manufactured industrially. Phenols as a class are less important today from the perspective of their being critical contaminants in groundwater, because there is no longer an OWDO or RUP limit for phenols (MOEE, 1992).

Phenol is a bipolar chemical with relatively high solubility, which is not easily adsorbed (Albaiges et al, 1986). Biotransformation can however be a significant attenuative mechanism for phenols. Chlorinated phenols (included in the halogenated hydrocarbon class) are used as wood preservatives, and can be significant contaminants in leachates from landfills which have received such wastes.

d) Carboxylic Acids

These are an important class of compounds which are found in nature. Carboxylic acids ionize in water and are quite soluble (Domenico and Schwartz, 1990). They are important components of leachate as a result of fermentation processes taking place under the anaerobic conditions inside landfills (Domenico and Schwartz (1990), and they may make up the majority of the organic compounds in leachate (Albaiges et al, 1986).

Some carboxylic acids such as the herbicides 2,4-D and 2,4,5-T are produced industrially and may be critical contaminants. Little information on the movement of carboxylic acids in leachate-derived groundwater contamination plumes was found in the published literature.

e) Polynuclear Aromatic Hydrocarbons

Polynuclear aromatic hydrocarbons (or PAHs) occur naturally, and are associated with ancient sediments and crude oils (Domenico and Schwartz, 1990). PAHs which may be critical contaminants in

landfill leachates include naphthalene (which is commonly found in leachates) and benzo(a)pyrene which has a very low ODWO and RUP limit.

The published literature about the mobility of PAHs in leachate-derived contaminant plumes is sparse, and thus it is difficult to identify critical contaminants from this class.

Non-Aqueous Phase Liquids (NAPLs)

The discussion of critical organic contaminants above has focussed on the attenuation in groundwater of dissolved organic contaminants, and has identified critical contaminants on the basis of the observed behaviour in groundwater of the dissolved contaminants in leachate-derived plumes.

There is however another possible mode of movement for organic chemicals in groundwater flow systems - the movement as a pure *non-aqueous phase liquid*, or NAPL. Many organic chemicals are not very soluble in water and may (if they are disposed of and subsequently escape from a landfill in sufficient quantities of pure chemical) move through the surrounding groundwater system as a discrete phase.

The behaviour in groundwater of NAPLs tends to depend on two parameters - their viscosity and their density. There is now an extensive body of information concerning the characteristics of "dense", or heavier than water, NAPLs (or DNAPLs) and of "light" NAPLs (or LNAPLs). Once they are present in an aquifer, both types of NAPLs can be long-term contaminant sources and can pose exceptional remedial problems.

DNAPLs tend to move downward through an aquifer until they encounter an impermeable unit, and then to move along the slope of the surface of the impermeable unit. Their movement is more or less independent of the groundwater flow directions in the aquifer. LNAPLs tend to float in a thin but laterally extensive layer on the water table at the top of the saturated portion of an aquifer.

The movement in aquifers of both types of NAPLs will not be affected by the attenuative mechanisms discussed in this paper, however dissolution of contaminants will occur from any NAPL source and the dissolved contaminants will be affected by attenuative mechanisms. Unfortunately, the rate of dissolution will often be so slow that a NAPL source can cause very long-term contamination of groundwater supplies.

5) Biological Contaminants

This type of contaminant includes pathogenic bacteria, viruses and parasites (Domenico and Schwartz, 1990). Research into the presence, mobility and persistence of biological contaminants in leachate plumes was not found in the published literature.

It is however not unreasonable to expect to find biological contaminants in landfill leachate plumes, because animal carcasses,

sewage sludge, and disposable diapers are all commonly disposed of at Ontario landfills.

While most pathogenic bacteria and parasites are relatively short-lived in groundwater flow systems, viruses may be much longer-lived and could cause contamination of groundwater supplies even if testing has shown that standard organic and inorganic contaminants do not exceed RUP limits.

6) Radioactive Contaminants

The wastes going into landfills are regulated by the province of Ontario through the MOEE, however radioactive contaminants come under federal and not provincial jurisdiction. Federal regulations do, however, specifically provide for the possibility of the disposal of low-level radioactive wastes in landfills and such disposal has occurred at Ontario landfills.

If radioactive contaminants are present in a landfill, then they will be present in the landfill leachate and they can be expected to become part of the groundwater contamination plume in the landfill's vicinity. Landfill groundwater monitoring programs generally do not include sampling for radioactive contaminants.

Fortunately, most radioactive contaminants (with the exception of tritium) tend not to be mobile under the hydrogeochemical conditions which prevail in groundwater systems across southern Ontario (which is where low-level disposal is likely to occur). It is interesting that tritium has been found at elevated levels in the Glenridge landfill in St. Catharines and the Borden landfill.

3.4 Discussion

Dilution is the most important attenuative process, because it affects all of the leachate-derived contaminants in a landfill's groundwater contamination plume. Although research into the migration of contaminant plumes has shown that a variety of attenuative processes occur and will effectively prevent the migration of some contaminants, other contaminants will be relatively unaffected by any attenuative processes except for dilution.

As a result, contaminant plumes in the vicinity of landfills can be expected to move at about the same rate as the groundwater itself. It is important to recognize however that individual contaminants may be effectively attenuated by processes besides dilution, and may as a result not be mobile in a landfill's contaminant plume. There is increasing research into the possibility that the addition of nutrients to aquifers may enhance the attenuation through biotransformation of specific organic contaminants (Acton and Barker, 1992), but considerable additional work is required in this field.

The list of critical contaminants in the vicinity of an attenuation landfill may include chloride, iron, manganese, aromatic hydrocarbons such as benzene, halogenated hydrocarbons such as trichloroethylene, as well as

carboxylic acids and PAHs. The attenuative processes are best understood for inorganic contaminants and for aromatic and halogenated hydrocarbons.

The current understanding of attenuative processes is not sufficient to allow accurate predictions to be made about the nature and extent of attenuation that will take place at a new landfill with the exception of very general comments. For existing landfills, the effect of attenuative processes can be more easily determined.

A common method of determining the net effect of all attenuative processes at an existing landfill is to compare the concentrations of various contaminants in the leachate to their respective levels in groundwater at increasing distances from the landfill. Case histories of four attenuation landfills where such work has been done are presented in Appendix D. Each landfill is located in a different hydrogeological setting in Ontario.

Attenuation in the surrounding groundwater flow system can be an appropriate method of dealing with the leachate from a landfill, provided that unacceptable off-site impacts are prevented from occurring. Given that dilution is the most important attenuative mechanism, a useful approach to the evaluation of attenuation landfills might be one which considers both the likely mass loading rates of contaminants to the groundwater flow system and the water balance for the landfill and surrounding area.

Appendix C of this review provides a more detailed discussion of suggested guidelines which might be used in the evaluation of prospective attenuation landfills.

4.0 The Fate of Leachate in Waste Water Treatment Plants

Landfill leachate is a strong, complex liquid for which treatment to meet surface water discharge criteria is a difficult task. There are several factors which contribute to this difficulty:

- Leachate flow changes dramatically during the treatment period; variability occurs both seasonally and in the longer term up to closure and beyond. Seasonal variations in leachate flow result from variations in temperature, precipitation, and evapotranspiration. Long term increases in flow are caused by increases in landfill surface area during operation. The provision of final cover will subsequently reduce the rate of final leachate production.
- Leachate contaminant concentrations change significantly over the treatment period as does the dominance of contaminant species which dictate treatment type and sequence.
- The breadth of contaminant types requiring treatment is greater than for most wastewaters.
- The high concentrations of certain contaminants require that, to meet the more stringent effluent discharge criteria, reductions up to five orders of magnitude are required. This is likely to be the case for ammonia, BOD, and iron.
- Predicting leachate flow and composition accurately enough to design an effective leachate treatment system at the time of landfill design is rarely possible, and predicting the time over which treatment will be required is even more difficult.
- Landfills are often not located near existing treatment facilities, since they are rarely located with treatment possibilities for leachate as a site location criterion.

These factors are more of a problem for landfill leachates than they are for most industrial wastewaters. However, as this report will show, excellent treatment of leachates is possible in most cases if proper design and treatment system operation is undertaken.

4.1 GENERAL TREATMENT OPTIONS

Landfill leachate treatment is most frequently undertaken in existing, waste water treatment plants (WWTP) and in many cases, this is the most cost effective form of treatment. Leachate is also occasionally pretreated on-site at the landfill to reduce sewer surcharge costs or to remove specific contaminants to a level where the leachate becomes acceptable for discharge to a sewer. Organic overload on the WWTP aeration system or the addition of heavy metals to the sludge discharge stream are frequently cited as reasons to regulate leachate discharge to existing facilities.

In many circumstances, although appropriate treatment process types and sequences exist at WWTP's, the key factor involved is dilution. If the leachate received at a community's WWTP has been generated from only the community's solid waste and if good cover has been placed on the landfill to reduce infiltration to a few centimetres per year, it is possible to achieve dilution factors at the WWTP in excess of 1:1000.

On-site leachate treatment to meet stream quality, final effluent discharge criteria (FEDC) is rare, although Canada is among the international leaders in this regard with two such facilities in operation: one near Halifax (Wright and Austin, 1988) and one in Sarnia (Lugowski and Poisson, 1991). In both cases, treated leachate is discharged to surface waters. In addition process sludges are discharged back to the landfill, an option which is not possible at some landfills in the United States where such sludges are defined as hazardous wastes.

4.2 CHARACTERISTICS OF LEACHATE AFFECTING TREATMENT

4.2.1 Flow Variations

Leachate flow from landfills is initially delayed as moisture infiltrating the surface is absorbed by the waste. The pore structure of the landfill is far too heterogeneous for the concept of field capacity throughout the landfill to be considered, however significant moisture is taken up prior to leachate being discharged at the base of the landfill. It is not uncommon to find leachate flow delayed for several years at sites where the vertical progression of the landfill is rapid. The large Keele Valley landfill is a case in point, however similar patterns are evident on a much smaller scale of waste depths of a few meters (Fungaroli and Steiner, 1984).

Seasonal variations in leachate flow are caused in part by changes in the rates of precipitation, runoff and evapotranspiration. Smaller landfills can experience near zero leachate flow in summer and late fall (McGinley and Kmet, 1984; and Robinson, 1987). Seasonal fluctuations in leachate flow diminish as the size and the age of the landfill increase.

The efficiency of most treatment processes is maximized when the flow is constant. Conditions where flows vary daily and weekly require over-design if peak or near peak flows are to be used for design. Seasonal flow variations where the lower and upper 25th percentiles differ by a factor of 2 or more may dictate that major process modifications such as temporary removal of equipment from service be undertaken. Low residence time, biological treatment processes such as biomass recycle units, biofilm units and sludge blanket processes, as well as chemical treatment processes are examples of processes where such modifications may be necessary.

Short term flow equalization in storage tanks is justified in some cases with constraints being tank size limitations and the development of biomass and/or precipitates within the tank.

Leachate recirculation within the landfill has the potential not only to dampen flow variations but also to provide significant improvement in leachate quality. The benefits of leachate recirculation have been shown convincingly at pilot scale (Pohland, 1987). However, full scale operational problems such as odours, precipitate formation, enhanced gas production and winter operation, plus concern for liquid head build-up within the landfill have generated concerns about recirculating leachate.

The annual average leachate flow at a landfill increases over the operational life of the site. In many cases this will require that installation of the treatment facility occurs in stages. Although highly variable, the average annual leachate flow is comparatively small at landfills where surface water and groundwater control is exercised and where good cover design exists.

4.2.2 Leachate Composition

The complexity of landfill leachate composition requires that a train of several unit processes be implemented to achieve adequate treatment. The capital and operational costs of treatment increase with increased process number and diversity. To achieve stream quality FEDC with a young landfill leachate, a functional process train might be that shown in Figure 1. Temperature control, especially to achieve nitrification, might also be necessary. Such a process sequence operating at a landfill facility with low leachate flows could conceivably be housed in a structure to facilitate higher treatment temperatures.

Typical leachate contaminant concentrations are shown in Table 3. The data are for units or cells of solid waste within a landfill and the time denotes the age of a specific cell. The composition of the total leachate can be obtained by calculating an age and flow weighted average strength over the whole landfill for each contaminant.

When considering the treatment of leachate, it is useful to discuss the leachate in terms of the following four categories:

- 1) organic matter
- 2) nitrogen
- 3) metals
- 4) other contaminants

1) Organic Matter

Biodegradable organic matter dominates leachate contaminants in the early years of leachate production. BOD₅ concentrations are

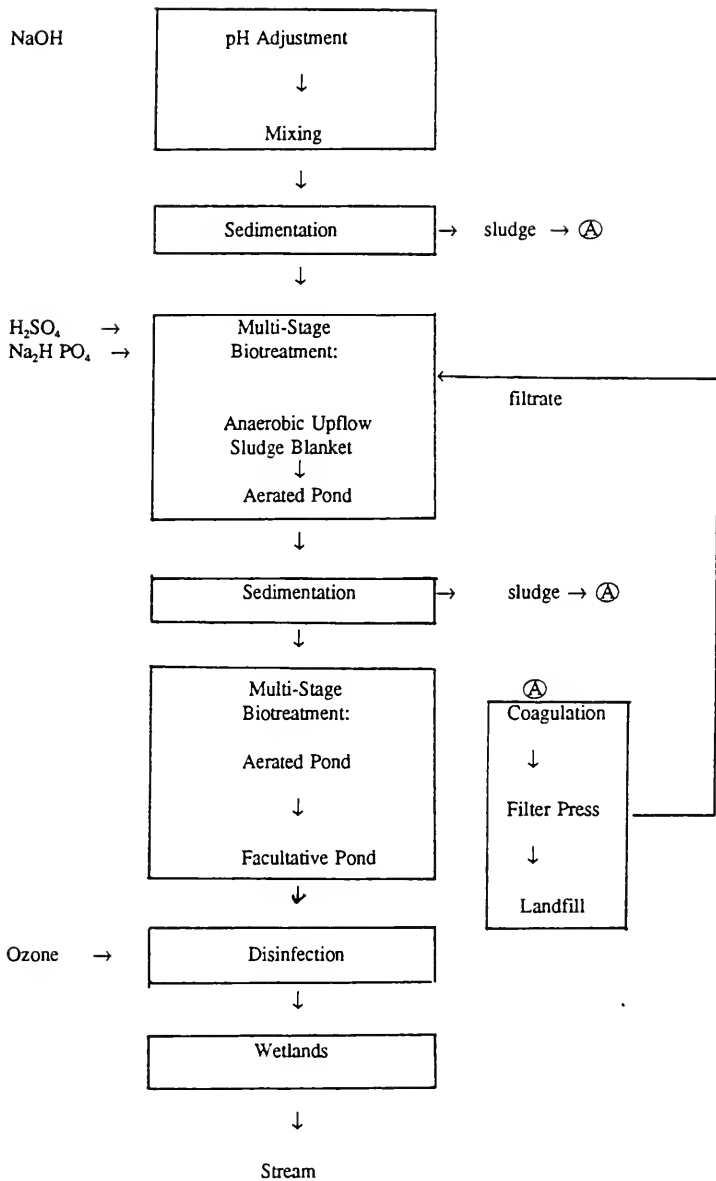


Figure 1: Possible Treatment Process Train to Achieve Stream Quality FEDC For a Young Landfill Leachate

frequently in the 10,000 to 20,000 mg/l range with the majority of this due to lower molecular weight volatile fatty acids (VFA). Acetic acid is the major form with significant amounts of propionic, butyric and caproic acids also present. These acids contribute to lower pH's in the 6.0 to 6.5 range for young leachates. Biological treatment of such leachates is essential but the problem is to decide which processes to use: aerobic versus anaerobic and fixed film versus suspended growth.

As the site ages, the BOD₅ of the leachate declines reaching levels in the 1,000 mg/l to 100 mg/l range after closure. Such concentrations can persist for many years after closure and will require biological treatment usually with an activated sludge or aerated pond configuration.

Leachate will also contain recalcitrant organics consisting mainly of humic material with traces of priority organics as well. Aromatic hydrocarbons such as benzene and toluene, and halogenated hydrocarbons such as trichloroethylene and methylene chloride are frequently present in leachate with concentrations in the high ug/l range (McGinley and Kmet, 1984). Treatment of recalcitrant organics, if needed, can be accomplished after biotreatment with carbon adsorption or oxidation. The ratio of BOD₅ to chemical oxygen demand (COD) tends to be in the range of 0.7 for young leachates. For old leachates several years after landfill closure, the BOD₅: COD ratios drop to the range of 0.1, consistent with the predominance of recalcitrant organics.

2) Nitrogen

Total Kjeldahl nitrogen (TKN) concentrations are generally in the range of 1,000 to 2,000 mg/l in young leachates with ammonia (NH₃-N) nitrogen, making up approximately 75% of the TKN. Much of the TKN will be taken up by the biomass in aerobic biotreatment processes in proportions of approximately BOD₅: TKN = 20:1. However, conditions of excess TKN are common and require that the aerobic systems be designed to achieve nitrification.

Unfortunately, TKN tends to remain at concentrations in the 50 to 200 mg/l range long after landfill closure. Since BOD₅ concentrations reduce more rapidly than TKN, there may be the need to supplement the biodegradable organic concentrations to maintain nitrification, the preferred removal process. Other TKN removal processes applied to landfill leachate include air stripping (Keenan, Steiner and Fungaroli, 1984) and oxidation. The former is very inefficient at low temperatures and may violate air quality standards. The latter becomes very costly as the TKN and the non-biodegradable carbon concentrations increase.

Table 3: Typical Landfill Leachate Contaminant Concentrations at Different Ages

PARAMETER	CONCENTRATION (mg/L) AT TIME (YR) AFTER PLACEMENT		
	1	5	15
BOD	20,000	2,000	50
TKN	2,000	400	70
AMMONIA - N	1,500	350	60
TDS	20,000	5,000	2,000
CHLORIDE	2,000	1,500	500
SULPHATE	1,000	400	50
PHOSPHATE	150	50	---
CALCIUM	2,500	900	300
SODIUM, POTASSIUM	2,000	700	100
IRON, MAGNESIUM	700	600	100
ALUMINUM, ZINC	50	50	20
COPPER, LEAD	<10	---	---
CADMIUM, MERCURY	<1	---	---

3) Metals

The dominant metals found in landfill leachates consist of the alkali earth metals, Na, Ca and Mg and to a lesser degree the heavy metals Fe, Zn and Pb. There are three main treatment problems presented by these metals:

1) Metals must be removed to meet FEDC; Fe is generally the dominant metal in this regard since removal from the high 100's mg/l to less than 1 mg/l is generally required.

2) Heavy metal loadings due mainly to Zn and Pb and to a lesser degree Cu may preclude the discharge of leachate to a WWTP if the disposal of treated sludge onto land is practiced.

3) Precipitate formation during leachate treatment is extensive due mainly to the formation of CaCO_3 and Fe_2O_3 . These precipitates coat surfaces and interfere with oxygen transfer and biomass attachment. They will also comprise a high percentage of the biological sludges.

The formation of Ca and Fe precipitates occurs when pH and Eh increase above the levels of the leachate upon discharge from the landfill. These increases accompany most leachate treatment and handling schemes and thus precipitate formation is inevitable. Full scale leachate treatment systems will benefit from Ca and Fe removal efforts at the influent end of the treatment system. Unit processes could include caustic soda (NaOH) addition to increase pH and preaeration to increase Eh accompanied by sedimentation to remove precipitates (Lugowski and Poisson, 1991).

Sludge generated by precipitation of metals with caustic addition is a hazardous waste in some jurisdictions, and disposal of the sludge in the landfill may therefore not be allowed.

4) Other Contaminants

The acidic pH of young leachates generated in the first few years of a landfill's life usually warrants neutralization. Neutralization results in removal of some metals and it increases pH to the neutral range which is suitable for leachate biodegradation. Neutralization with lime (CaO) is not advisable because much of the Ca added precipitates as CaCO_3 , adding to the already large volume of sludge produced and to scale formation in pipes, on aeration devices and elsewhere throughout the treatment scheme.

The total dissolved solids (TDS) of treated leachates are generally high and this is often associated with foaming at the point of discharge as a result of higher surface tension. If

treatment is required it will likely involve a reverse osmosis (RO) process. Adequate disposal of the concentrate from the RO process can make this a very costly treatment scheme (Logemann, 1990).

The total phosphate (PO_4) concentration in treated leachates are generally high and this is often associated with foaming as a result of higher surface tension. If treatment is required, it will likely involve PO_4 supplement.

4.3 EFFECTIVENESS OF SPECIFIC TREATMENT PROCESSES

The strength of landfill leachate as compared to typical FEDC and the diversity of contaminant types to be removed require that sequences of treatment processes be used to achieve adequate treatment. This section examines the features of specific process types that have been used to treat leachate.

4.3.1 Biological Treatment Processes

A biological treatment process should be present in a landfill leachate treatment train for as long as there is sufficient biodegradable organic matter to support it. There are many reasons why biological treatment processes should be present:

- Biotreatment processes actually destroy organic contaminants rather than shunt them from one phase or process stream to another. This destruction in most cases will involve conversion of organic contaminants to CO_2 , H_2O , NH_3 , and other much simpler. It will also, however, result in the production of additional biomass especially in aerobic processes which will require treatment and disposal.
- Bioprocesses frequently increase pH and aerobic processes increase Eh as well. Both result in significant metal removal with precipitates becoming part of the biosludge. Certain recalcitrant organic compounds (mostly hydrophobic compounds) will adsorb on to the biomass resulting in removal from the liquid phase.
- Aerobic processes can be designed to nitrify, resulting in the conversion of ammonia to nitrate, and improving the final effluent discharge quality by approximately three orders of magnitude.
- Aerobic processes can be used to air-strip volatile organic contaminants.
- Biotreatment processes are often less costly in terms of \$ per cubic metre treated than other processes.

There are several types of biological treatment processes and it is often difficult to decide which process to use and in what sequence. For example, the most common biological treatment process options include the following:

- aerobic versus anaerobic processes
- suspended versus attached processes
- recirculated versus single pass processes

The following discussion considers 1) aerobic and 2) anaerobic processes with respect to leachate treatment for various landfill conditions. Leachate derived from municipal solid waste is seldom sufficiently inhibitory to preclude either aerobic or anaerobic biotreatment systems, thus other factors, which are outlined below, are used to determine which type of system is used.

1) Anaerobic Processes

Anaerobic processes are justified for the treatment of high strength leachates with BOD's in excess of 1,000 mg/l. The elimination of aeration costs, the production of CH_4 to heat the reactor, lower biomass production and excellent conversion of BOD from the 10,000's mg/l to the low 100's mg/l are reasons to consider anaerobic processes. Packed bed attached biomass (PBR) and upflow anaerobic sludge blanket (UASB) systems have been the only anaerobic systems used successfully at full scale in North America. A PBR has been in operation at the Omega Hills landfill in Milwaukee, Wisconsin for several years (Schafer *et al.*, 1987). A PBR was used initially at the Highway 101 landfill in Halifax but this was subsequently converted to an UASB (Wright and Austin, 1988). The conversion was necessary because of clogging due to CaCO_3 and FePO_4 precipitation on the plastic filter medium. This is likely to be a problem with most young leachates and will encourage the use of UASB units. The experience with UASB at the Highway 101 landfill has been positive.

In order to be efficient, anaerobic processes require temperatures in the range of 30°C to 35°C. This is achieved with heat exchangers using water heated with CH_4 derived from the process itself. Experience with both PBR and UASB systems has shown good pH adjustment, TDS reduction, metals removal and some removal of recalcitrant organics in addition to excellent BOD reduction.

The pattern observed at many landfills is the production of strong leachates with BOD's in the 10,000's mg/l for the first few years, followed by BOD's in the range of 1,000 mg/l to 5,000 mg/l for many years. This latter concentration range is marginal for the efficient use of anaerobic processes. As a result, the viability of anaerobic processes may be limited to the first few

years of leachate production.

Effluent from anaerobic leachate treatment processes is characterized by ammonia and BOD in the 100's of mg/l. This is suitable for discharge to a WWTP as at the Omega Mills landfill (Schafer et al., 1987) but not for discharge to surface waters. If discharge to surface waters is required, then further treatment involving aerobic processes is needed for nitrification and additional BOD removal.

2) Aerobic Processes

Depending on the leachate quality, aerobic biotreatment processes can be used either a) following anaerobic processes, b) in conjunction with physical-chemical processes or c) used alone. They are essential in the treatment of leachate from municipal solid waste landfills if surface water-based FEDCs are to be met for BOD, ammonia ($\text{NH}_3\text{-N}$), and iron. Although exceptions will always occur, the information given in Table 4 serves as a guideline for the selection of aerobic processes.

It is expected for all processes that pH will have been neutralized and that PO_4 will have been supplemented. All processes could be designed with either BOD removal or nitrification controlling, although the latter is expected to be the case for leachates with low BOD. BOD supplement may be necessary to achieve satisfactory oxidation of ammonia in old leachates.

4.3.2 Physical-Chemical Treatment Processes

The early work of Chian and DeWalle (1977) showed that physical-chemical treatment processes alone are unlikely to provide adequate and cost-effective treatment of landfill leachates. Certain physical chemical processes are however useful, and often essential, in conjunction with biological processes. The most likely processes in this regard are summarized in the following sections although some have been mentioned previously.

1) Chemical Addition and Sedimentation

Chemical addition and sedimentation is likely to be required as first stage of treatment for landfill leachates. NaOH will raise the pH for improved biotreatment and will result in the precipitation of metals, primarily Ca and Fe. The largely metallic sludge produced in this process sequence will consist mainly of CaCO_3 , Fe_2O_3 and $\text{Fe}_2(\text{CO}_3)_3$, but other heavy metals will also be precipitated. Disposal of the sludge could be undertaken

Table 4: Guidelines For Selecting Aerobic Biotreatment Processes

A. Aerated Pond (No Biomass Recycle)

- BOD above 500 mg/l to maintain adequate biomass concentrations.
- Effluent biomass sedimentation and disposal required.
- Poor operation in cold temperatures especially with respect to nitrification.

B. Activated Sludge or Sequencing Batch Reactors (Biomass Recycle)

- BOD below 1,000 mg/l
- Wasted biomass requires disposal
- Lower HRT's reduce the impact of cold weather operation

C. Facultative Ponds (No Biomass Recycle)

- BOD and $\text{NH}_3\text{-N}$ less than 50 mg/l and 25 mg/l respectively.
- Large area requirement.
- Near 0°C temperatures in winter necessitate seasonal discharge.
- Periodic clean out required.

D. Rotating Biological Contactors and Aerobic PBR

- BOD below 200 mg/l.
- Heating/insulation required in winter conditions.
- Effluent sedimentation is required.

in several ways, but dewatering and burial in the landfill are likely to be the most cost effective methods.

If required, metal precipitation may be increased by raising the pH to 9, but subsequent acid addition will then be needed to reduce the pH to 7.5 for biotreatment. Sedimentation to remove metallic precipitates will follow. Prior addition of coagulants and coagulant aids may be necessary to enhance precipitation, floc formation, and sedimentation.

2) Oxidants and Pre-Aeration

The addition of chemical oxidants such as H_2O_2 , O_3 , and $KMnO_4$ may be warranted for colour removal, odour control, and also to enhance the removal of multivalent metals including Fe. Pre-aeration could provide similar treatment.

A process sequence might include the following steps:

Pre-aeration or --> NaOH Addition --> Flocculation
Oxidant Addition and Mixing

--> Sedimentation and
Sludge Disposal

3) Granular Filtration

Granular filtration units are designed to remove suspended solid (SS) and would therefore be used prior to effluent discharge or as pretreatment for granular carbon adsorption or a membrane process such as reverse osmosis. Disposal of backwash water which might represent up to 10% of the forward flow would be required.

The City of Sarnia Leachate Treatment Facility includes granular filtration prior to discharge to effluent treatment lagoons (Lugowski and Poisson, 1991).

4) Granular Carbon Adsorption

Granular carbon adsorption columns would be used near the end of a leachate treatment process train for the removal of recalcitrant organic matter. However, the need for such systems has been rare in leachate treatment schemes reported up to this time.

5) Membrane Processes

Membrane processes and reverse osmosis in particular, have the potential to separate leachate into a high flow, low contaminant

stream called permeate and a low flow, high contaminant stream called concentrate. While separation efficiencies can be high, so can the costs, especially those related to the treatment and disposal of the concentrate stream (Logemann, 1990).

4.4 CASE HISTORIES OF FULL-SCALE LEACHATE TREATMENT

Examples of full-scale MSW landfill leachate treatment systems are rare in North America, and certainly in Canada. The facility at the Highway 101 Landfill near Halifax, Nova Scotia was initiated in 1987 and was the first major leachate treatment system in Canada (Wright and Austin, 1988). The facility at the Sarnia Landfill implemented in 1990, was the first full-scale, on-site leachate treatment system in Ontario (Lugowski and Poisson, 1991). In both cases, the treated effluent is being discharged into surface waters.

Appendix A contains descriptions in summary of full-scale systems treating MSW landfill leachate only (A.1) and MSW/hazardous waste leachate (A.2). The process treatment trains and the removal efficiencies vary significantly from one application to another because of variations in site age, climate and FEDC. However, similar solutions exist for similar problems and these are, in general, consistent with the information presented in this paper.

4.4.1 Highway 101 Landfill (Wright and Austin, 1988)

The Highway 101 Landfill leachate treatment system is examined in more detail to highlight some of the principles and problems common to most MSW leachate treatment schemes.

1) Leachate Characteristics

In the mid-1980's as the treatment system was being studied at pilot-scale, the leachate flow was approximately 80 m³/d on average, although the design flow used was a factor of approximately 3 greater than this value. Typical leachate strength at that time was:

COD	=	22,800 mg/l	Fe	=	940 mg/l
BOD ₅	=	16,100 mg/l	Zn	=	70 mg/l
TOC	=	8,100 mg/l	Ca	=	1,740 mg/l
NH ₃ -N	=	400 mg/l	Cl	=	1,110 mg/l
pH	=	5.6 mg/l	TDS	=	15,300 mg/l

In 1990, the strength of the leachate had reduced significantly with some COD measurements less than 10,000 mg/l and reduced NH₃-N and Fe concentrations. These are consistent with expectations as a landfill ages.

2) Final Effluent Discharge Criteria (FEDC)

The FEDC are presented in detail by Wright and Austin (1988) and not reproduced here. However, they were based upon requirements for discharge to the Sackville River.

3) Flow Equalization

Some equalization of flow was achieved through the use of a 50 m reinforced concrete tank with a design flow hydraulic retention time of approximately 5 hours.

4) Pre-Treatment

Pre-treatment consisted of NaOH addition, flash mixing, flocculation and sedimentation to raise the pH to neutrality and precipitate metals. Pilot scale test results showed pH adjustment to 7.2, Fe reduction to 155 mg/l and Zn reduction to mg/l.

Subsequent problems were experienced with extensive scale buildu on biofilter media. This was due mainly to Ca and Fe precipitat formation. It might have been better to raise the pH to 9 in this process for greater removal of metals and then to neutraliz to pH 7.5 with H_2SO_4 . Settled sludges were discharged to the landfill.

5) Anaerobic Treatment

The pilot-scale experiments were performed with a PBR anaerobic filter heated to 32°C. BOD₅ reductions from 16,000 mg/l to 260 mg/l exhibited outstanding treatment efficiency. The full-scale unit provided similar efficiency but problems developed with scale formation on the filter medium. The PBR was subsequently replaced with an UASB system. Excellent BOD₅ destruction has been achieved.

6) Aerated and Facultative Ponds and Wetlands

The function of the aerated and facultative ponds was to improve removal of biodegradable organics and suspended matter and to achieve nitrification. They have performed well in this regard especially with respect to meeting FEDC BOD requirements for the Sackville River. Ammonia removal has been good but the effluent from the ponds does not meet the FEDC. As a result, the pond effluent has been discharged to natural wetlands with in turn drain into the Sackville River. Data from November, 1991, presented below show a modest increase in ammonia nitrogen in th Sackville River as a result of discharge from the wetlands.

Ammonia Nitrogen Concentrations in the Sackville River

<u>Date</u>	<u>Above Wetlands</u>	<u>Below Wetlands</u>
91.11.07	<0.05 mg/l	<0.05 mg/l
91.11.13	0.21 mg/l	0.22 mg/l
91.11.21	<0.05 mg/l	0.1 mg/l
91.11.27	<0.05 mg/l	0.08 mg/l

4.5 DISCUSSION

Research and field experience has shown that very good treatment of landfill leachate is possible whether in a WWTP or in an on-site facility treating undiluted leachate. Stream quality FEDC have been achieved at on-site facilities in spite of many initial contaminant concentrations exceeding 100x typical values reported for municipal sewage.

The on-site treatment systems have been complex, although no more so than the advanced systems found at certain Ontario WWTP such as the one in Guelph. If required, metal precipitation may be increased by raising the pH to 9, but subsequent acid addition will then be needed to reduce the pH to 7.5 for biotreatment. Sedimentation to remove metallic precipitates will follow. Prior addition of alum to assist coagulation and flocculation may be needed.

The discharge end of the Guelph system includes biodisks to enhance BOD removal and nitrification and filters for solids removal. This tertiary-level treatment would be needed at most on-site leachate treatment facilities.

One major concern that persists is the high level of recalcitrant organic matter in effluents from on-site leachate treatment systems. Typical effluent COD measurements are in the mid-100's range. This has been attributed mainly to humic material as opposed to harmful organics, however concern exists for the fate of this material during chlorination (the disinfection process used at most treatment facilities). The formation of chlorinated organics is expected to be significant, although the types of products formed and their toxicity have not been established. The use of ozone instead of chlorine or discharging of the treated effluent to a wetlands environment as is being done at the Sarnia facility will avoid the problem.

The high level, on-site leachate treatment needed to meet stream quality FEDC extracts a high price. The cost to construct the Highway 101 Landfill leachate treatment system in 1988 was approximately \$3.0 million; the cost to operate and maintain it in 1988 was estimated to be \$155,000 per year. The cost to complete the Sarnia Landfill leachate facility in 1990 was approximately \$2.0 million with operating and maintenance costs estimated at \$100,000 per year.

Most landfills will produce leachate forever. It is not possible to

predict how long the need to treat the leachate will persist, but it is assumed to be for several decades after closure. Consequently, long-term treatment costs will result. However, the need for certain processes will be limited and simplification to the treatment scheme and a lowering of treatment costs will result.

There are situations in which leachate treatment will not be feasible or warranted:

1) Landfill sites serving rural communities may not require costly leachate treatment if adequate attenuation is provided in the groundwater flow system in the vicinity of the landfill. It is difficult to anticipate the size and conditions of a landfill site where treatment would become necessary. Well operated sites of a few hectares in size located well away from major aquifers are least likely to require treatment. One difficulty may arise at small sites where the permeability of the underlying soils is too low and the leachate does not drain adequately to the subsurface. Surface leachate discharge resulting at such sites will require some form of remediation which may involve treatment.

2) The treatment facilities in the region of a landfill may not be suitable to receive leachate. Decisions in this regard will depend on an assessment of contaminant loading from the landfill, treatment-dilution capacity of the facility, and discharge constraints placed on it for both liquid effluents and sludges. Facilities with primary treatment only would not likely be suitable to receive leachate unless dilution were substantial.

3) Facilities which dispose of digested sludges on land will be regulated by metals loadings; metals input from leachate will be of concern at such facilities.

4) In general, treatment processes can be designed for WWTPs to handle any landfill's leachate. The major limitations are the FEDC for the receiving surface water body, and the costs associated with achieving adequate levels of treatment such that leachate will meet the FEDC. Difficulties in achieving the FEDC, particularly in small or sensitive surface waters, may at some locations make leachate treatment unfeasible from a cost perspective.

5.0 The Relative Merits of Using Groundwater Attenuation versus Using Waste Water Treatment Plants to Deal with Landfill Leachates

Generally, the trend in the evolution of Ontario landfill designs has been away from attenuation landfills toward containment landfills. There appear to be three main reasons for this trend:

- the difficulty in controlling and predicting leachate migration and impacts in groundwater at attenuation landfills;
- different regulations and standards of enforcement for landfill discharges to groundwater vs. WWTP discharges to surface water;
- the political problem of "selling" the design concept of an attenuation landfill to a mistrustful public.

Nonetheless, there may be situations where an attenuation landfill will provide a more appropriate or effective method of leachate treatment than a containment landfill.

Attenuation in groundwater may tend to be more attractive as the preferred method of dealing with a landfill's leachate under combinations of one or more of the following circumstances:

- if the site hydrogeology is favourable for the development of an attenuation landfill;
- if the landfill is small;
- if WWTPs which can adequately deal with the landfill's leachate are not available in the surrounding area (in our opinion, WWTPs providing only primary treatment are generally not suitable for treatment of landfill leachates);
- if it can be demonstrated that the net costs and environmental impacts of leachate attenuation in groundwater are less than the net costs and impacts of leachate collection, treatment in a WWTP, and discharge to surface waters.

If, at a given location, consideration is being given to a proposal to establish an attenuation landfill then the suggested Guidelines for evaluating such proposals, which are outlined in Appendix C of this paper, may prove helpful.

Similar guidelines should, in our opinion, be prepared for evaluating proposals to collect and treat landfill leachate at a nearby WWTP. Such guidelines should include the following minimum requirements:

- the lifespan of the landfill's leachate collection system should exceed the contaminating lifespan of the landfill;
- the hydrogeology of the landfill should be well understood and contingency groundwater treatment measures should be prepared, in the event of failure of the leachate collection system;

- the WWTP should meet the applicable Ontario FEDC (either MISA or the PWQO) at either the point of discharge to surface waters, or at the property boundary if the point of discharge is on-site;
- the WWTP should be regularly monitored to ensure compliance with the FEDC;
- contingency plans should be established to cover the possibility of a breakdown of the WWTP, and such plans should meet the same minimum requirements set out for contingency plans in Section 3.5.

Leachate collection and treatment will tend to be more attractive as the preferred method of dealing with a landfill's leachate under combinations of one or more of the following circumstances:

- if the landfill is very large;
- if WWTPs which can adequately deal with the landfill's leachate are located within a reasonable distance from the landfill;
- if the local groundwater resource is a good quality aquifer;
- if it can be demonstrated that the net costs and environmental impacts of leachate collection, treatment in a WWTP, and discharge to surface waters are less than the net costs and impacts of leachate attenuation in groundwater.

The current process for evaluating proposals to establish or expand landfills may not provide a balanced approach to the consideration of the two methods of dealing with a landfill's leachate. This is because there are different regulations and standards for enforcement for landfill discharges to groundwater versus WWTP discharges to surface water. In addition, the treatment of leachate in a distant WWTP is generally not considered at hearings for landfills while the treatment of leachate by attenuation in the nearby groundwater flow system is rigorously evaluated at such hearings. It seems reasonable that the net environmental impacts of the chosen method of dealing with a landfill's leachate be evaluated with equal rigour for both attenuation landfills and for landfills where leachate will be collected and treated at a WWTP.

6.0 CONCLUSIONS AND RECOMMENDATIONS

- 1) There are generally both spatial and temporal variations in leachate strength, leachate chemistry, and in rates of leachate generation at a given landfill which must be considered in the evaluation of proposals for both attenuation and containment landfills.
- 2) Attenuation landfills will have a greater impact on groundwater (since the local groundwater flow system is being relied upon for leachate treatment), while containment landfills will have a greater impact on surface waters (because this is where the effluent, after leachate is treated at a WWTP, is being discharged).
- 3) The most important process of attenuation in groundwater flow systems is dilution, because it affects all contaminants in a leachate plume.
- 4) Although various attenuative processes may effectively prevent the migration of some contaminants in a leachate plume, other contaminants will be relatively unaffected by processes other than dilution.
- 5) Given the importance of dilution as an attenuative mechanism, the evaluation of proposed attenuation landfills should include explicit consideration of the local water balance, including consideration of the volume of groundwater available for dilution and likely mass loading rates for critical contaminants in the landfill's leachate.
- 6) Trace metals are unlikely to be mobile at the neutral pH levels found in southern Ontario groundwaters, thus they are unlikely to be critical contaminants at attenuation landfills.
- 7) Of the various inorganic parameters chloride, iron, and nitrate are most likely to be critical contaminants at attenuation landfills. Ammonia and various heavy metals may pose problems if leachate is discharging to surface waters in the immediate vicinity of the landfill.
- 8) Many halogenated hydrocarbons may be attenuated through biodegradation under the anaerobic conditions found in leachate plumes. The biotransformation of halogenated hydrocarbons may lead to the creation of "breakdown products" which can pose special problems at attenuation

landfills. Of the halogenated hydrocarbons, the aliphatic halogenated hydrocarbons such as trichloroethylene and the aromatic halogenated hydrocarbons such as chlorobenzene are most likely to be critical contaminants at attenuation landfills.

- 9) Aromatic hydrocarbons are readily attenuated through biodegradation in aerobic groundwater conditions, but may pose problems under the anaerobic conditions found in many leachate plumes. Benzene is the aromatic hydrocarbon which is most likely to be a critical contaminant at attenuation landfills.
- 10) There is little if any published information which would allow a determination of possible critical contaminants in leachate-derived groundwater contamination plumes for the following classes or organic chemicals: phenolic compounds, carboxylic acids, and polynuclear aromatic hydrocarbons (PAHs). Further research is needed in this area.
- 11) Information pertaining to the presence in landfills of biological or radioactive contaminants, and of their attenuation in leachate-derived contamination plumes, was not found in the published literature. It is possible or even likely that biological and/or radioactive contaminants are found in leachate from Ontario landfills, and research is required to determine which are the possible critical contaminants for these groups.
- 12) Attenuation in the surrounding groundwater flow system can be an appropriate method for dealing with the leachate from a landfill.
- 13) Guidelines for evaluating proposed attenuation landfill sites have been prepared and are presented in Appendix C of this paper.
- 14) Further research into attenuation processes at existing landfills is recommended. In particular, a study of landfill case histories based on a review of consultants reports and monitoring data from existing attenuation landfills will prove helpful in determining likely critical contaminants from such sites.
- 15) Research and field experience has shown that very good treatment of landfill leachate is possible in both WWTPs and in on-site facilities treating undiluted leachate.
- 16) In general, treatment processes can be designed to handle any landfill's leachate, however at some locations the costs of providing adequate leachate treatment may make a containment landfill uneconomical.
- 17) Waste water treatment plants which provide only primary treatment are

generally not likely to provide adequate treatment of landfill leachates.

- 18) Physical-chemical treatment processes alone are unlikely to provide adequate and cost-effective treatment at WWTPs which receive landfill leachates.
- 19) Biological treatment should be included in the treatment train at a WWTP which receives leachate for as long as there is sufficient biodegradable organic matter to support it, because biological treatment will actually destroy organic contaminants and because it may also result in significant removal of both metals and recalcitrant organic compounds.
- 20) There are few examples of dedicated landfill leachate treatment systems in Canada - the facility at the Sarnia Landfill was the first full-scale, on-site leachate treatment system in Ontario. The treated effluent from this facility is being discharged into surface waters. Tertiary treatment of effluent will be needed at most on-site leachate treatment facilities.
- 21) On-site treatment facilities will be expensive to develop and to operate. It is not yet clear how long modern landfill leachates will require treatment, but the contaminating lifespan during which leachate at containment landfills must be collected and treated is likely to measured in decades. Over time following the closure of a landfill, the processes needed to treat its leachate can be simplified resulting in a lowering of treatment costs.

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